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The effectiveness of active and passive restoration on recovery of indigenous vegetation in riparian zones in the Western Cape, South Africa: A preliminary assessment

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ABSTRACT

Riparian ecosystems in South Africa's fynbos biome are heavily invaded by alien woody plants. Although large-scale clearing of these species is underway, the assumption that native vegetation will self-repair after clearing has not been thoroughly tested. Understanding the processes that mediate the recruitment of native species following clearing of invasive species is crucial for optimising restoration techniques.

This study aimed to determine native species recovery patterns following implementation of different management interventions. We tested the influence of two clearing treatments ("fell & remove" and "fell & stack burn") on the outcomes of passive restoration (natural recovery of native riparian species) and active restoration (seed sowing and planting of cuttings) along the Berg River in the Western Cape. Under greenhouse conditions we investigated seed viability and germination pre-treatments of selected native species.

There was no recruitment of native species in sites that were not seeded (passive restoration sites), possibly because of the dominance of alien herbaceous species and graminoids or the lack of native species in the soil-stored seed bank. Germination of our targeted native species in the field was low in both "fell & remove" and "fell & stack burn" treatments. However, "fell & stack burn" gave better germination for the species *Searsia angustifolia*, *Leonotis leonurus* and *Melianthus major*. Seedling survival in the field was significantly reduced in summer, with drought stress being the main cause for seedling mortality. Germination rates in the greenhouse were high, an indication that harvested seeds were viable. Most seeds germinated without germination pre-treatments.

We conclude that failure of native seeds to germinate under field conditions, secondary invasion of alien herbs and graminoids, the lack of native species in the soil-stored seed bank, and dry summer conditions hamper seedling establishment and recovery on sites cleared of dense stands of alien trees. For active restoration to achieve its goals, effective recruitment and propagation strategies need to be established.

1. Introduction

Riparian habitats provide many ecosystem services, including riverbank stabilisation, nutrient cycling, flood attenuation, regulation of streamflows and stream temperatures, groundwater recharge and water purification (Richardson et al., 2007). However, natural and human-related disturbances occurring along riparian systems have facilitated their invasion by alien plants (Richardson et al., 2007). Alien species diversity and abundance have increased in riparian systems worldwide (Hood and Naiman, 2000; Richardson et al., 2007). Most rivers in South Africa's fynbos biome are lined by dense stands of Australian *Acacia* and *Eucalyptus* species (Forsyth et al., 2004; Richardson and Van Wilgen, 2004; Meek et al., 2010, 2013). These invasions have displaced native species (Richardson et al., 1997; Richardson and Van Wilgen, 2004) and have caused significant changes to both above- and below-ground (seed bank) vegetation composition and guild structure (Vosse et al., 2008). Furthermore, alien tree invasions have substantially reduced streamflow (Dye and Poulter, 1995).

The Working for Water Programme (WfW) was established in 1995 to reduce the impacts of alien species in South Africa. One objective of this programme is to protect and maximise water resources by controlling invasive alien plants (Van Wilgen et al., 1998). Several studies have shown that streamflow increases after the removal of alien tree stands (Dye and Poulter, 1995; Prinsloo and Scott, 1999), but the extent to which native species recover after the removal of the alien trees is variable (Galatowitsch and Richardson, 2005; Blanchard and Holmes, 2008; Pretorius et al., 2008). There is an urgent need to improve our understanding of the impacts of clearing and the factors that influence the subsequent recovery of native species (Holmes et al., 2008).

Little attention has been given to deciding which removal strategy is not only most successful and practical, but also best in promoting natural (unassisted) native species recovery (passive restoration). A study by Blanchard and Holmes (2008) on Australian *Acacia* species in the

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mountain stream and foothill reaches of different rivers in the fynbos biome identified “fell & removal” as the best method for clearing stands of invasive species to facilitate the recovery of indigenous vegetation. On the other hand, burning is known to reduce the abundance of alien species, whilst also stimulating the germination of indigenous fynbos species (Blanchard and Holmes, 2008). But, fire also stimulates germination of alien species which potentially hinders restoration initiatives (Holmes et al., 2008). WfW teams typically fell alien trees and stack slash before burning it after allowing it to dry. Where necessary, herbicide is applied to the stumps to prevent the alien trees from resprouting. Although these clearing treatments are widely applied, their effectiveness has yet to be tested scientifically. The first aim of our study was thus to test the effectiveness of the two clearing treatments used by WfW, namely “fell & remove” and “fell & stack burn”, in promoting natural (unassisted) native species recovery.

Currently, WfW assumes that indigenous vegetation will “self-repair” and that ecosystems will be set on a trajectory towards restoration of pre-invasion structure and function once the main stressor (dense stands of alien invaders) has been removed (Esler et al., 2008). However, studies have shown that it takes several years for passive restoration to be successful mainly due to secondary invasion (Reinecke et al., 2008), resource alteration (Galatowitsch and Richardson, 2005) or ‘legacy effects’ — long-lasting changes in ecosystem structure (Holmes et al., 2008; Le Maitre et al., 2011). More recent research has shown that passive restoration may be difficult to achieve where key biotic and abiotic thresholds have been crossed and resilience has been reduced (Le Maitre et al., 2011; Gaertner et al., 2012); this is most likely in sites where dense invasive stands have been present for several decades (Holmes et al., 2008). This has led to suggestions that active restoration is needed when dealing with heavily invaded sites where thresholds have been passed (Holmes et al., 2008; Gaertner et al., 2012). However, very few studies have examined the effectiveness of active restoration in riparian systems.

Active restoration includes additional restoration interventions beyond removal of the invader so as to facilitate recovery (Holl and Aide, 2011). Such interventions are expensive, but because of the perceived benefits, several options have been tested in riparian ecosystems (Holmes et al., 2008). These include reintroducing propagules of native plants or animals, soil manipulations after alien removal and the active manipulation of disturbance regimes such as fire and flooding (Holmes et al., 2008). To our knowledge, only one study has examined the effectiveness of active restoration in riparian ecosystems in the Western Cape. This study looked at the effectiveness of sowing a mixture of seeds of indigenous plant species in restoring riparian vegetation (Pretorius et al., 2008). In this case the observed presence of native vegetation, eight years after the initial sowing, pointed to the potential of active restoration to facilitate recovery of native vegetation after alien removal.

Two of the commonly used planting techniques in active restoration include direct seeding and the transplanting of seedlings (Doust et al., 2008). Advantages and disadvantages of these techniques have been extensively studied under greenhouse conditions. However, only a few studies have tested these methods under field conditions. Propagated plants have been used simply because they establish more rapidly and increase the chances of restoration success; however they are costly and labour intensive. To our knowledge, no study has examined the direct introduction of cuttings in the field, as a less expensive technique compared to propagating such cuttings (or seedlings) in the greenhouse. Therefore, the second aim of our study was to determine the patterns of early native species recovery following seeding and planting of cuttings.

Some of the challenges faced in active restoration programmes include granivory or herbivory where restoration sites are not enclosed (Iponga et al., 2005) and the failure of native species to germinate due to dormancy (Florentine et al., 2011). Several studies have shown that seed predators, particularly herbivores and granivores, have the potential to significantly reduce seed germination (Crawley, 1992; Milton, 1995). Although seed burial reduces predation thereby enhancing seed survival and germination chances, Christian and Stanton (2004) showed that deeper burial can cause delayed seed emergence. To increase chances of seed germination, several seed pre-treatments for breaking dormancy and accelerating germination have been suggested (Budy et al., 1986). Our third aim was to test seed germination under various pre-treatments in the greenhouse. This is one of the few studies to test various germination treatments for fynbos species targeted for restoration (but see Brown and Botha, 2004).

To achieve our aims we addressed the following questions: (1) Which clearing method is most effective for promoting natural (unassisted) recovery of native species (passive restoration)? (2) How effective is active restoration (by means of seeding and cutting planting) for restoring indigenous vegetation following two treatments for removing stands of the invasive tree *Eucalyptus camaldulensis*: fell & remove and fell & stack burn? (3) Were seeds of introduced native species viable and which germination pre-treatment is appropriate for each of them?

2. Methods

2.1. Study site

The study area was situated along the Berg River in South Africa's Western Cape Province (Fig. 1). The river, approximately 294 km long with a catchment area of about 7 715 km^2, flows into the Atlantic Ocean at Velddrif (de Villiers, 2007). The geology of the upper Berg River catchment is dominated by sandstone and quartzites of the Cape supergroup, whereas the rest of the catchment is underlain by Cape granites and Malmesbury shale (de Villiers, 2007). The catchment is characterised by nutrient-poor lithologies, but some areas consist of deep alluvial flood plains with fertile sediments (de Villiers, 2007). Almost 50% of the catchment area is cultivated agricultural land. River flow peaks during the winter rainy season, from June to August, with rainfall averaging between 300 and 600 mm per annum. The part of the river where the study was conducted is located in the renosterveld (Mucina and Rutherford, 2006). Although fire plays an important role in shaping vegetation communities in the renosterveld (Van der Merwe and Van Rooyen, 2011), riparian vegetation along rivers like the Berg rarely burns. The small area of the remaining native vegetation along the Berg River is dominated by typical riparian species of the region, including *Kiggelaria africana*, *Olea europaea*, *Melianthus major* and *Searsia angustifolia* (Geldenhuys, 2008). The whole river stretch is heavily invaded by alien trees, mainly *E. camaldulensis*, with less abundant stands of other invasive alien plants, notably *Acacia longifolia*, *A. mearnsii* and *Populus* species (Tererai et al., 2013). Invasion of the Berg River by *E. camaldulensis* appears to have started about 50 years ago, but little is known about the early stages of invasion of the river (Geldenhuys, 2008). Also, no studies have reported on the pre-invasion conditions of the Berg River. Further details of the study sites are provided by Ruwanza et al. (2013a).

2.2. Passive and active restoration experiments

To examine the efficacy of both passive and active restoration, twelve sites representing four treatments (each replicated three times), namely two clearing treatments of fell & remove (F&R) and fell & stack burn (F&SB) as well as two control treatments of invaded (IS) and natural sites (NS), were selected. These were set up in the dry bank of the Berg River as the wet bank was very narrow. Prior to clearing, our sites (F&R and F&SB) were heavily invaded by *E. camaldulensis* (>75% canopy cover). In F&R, cut alien trees were removed from the riparian zone using heavy harvesting machines whilst in F&SB the cut alien trees were stacked and left to dry before being burnt. Clearing was completed in December 2010 and burning was conducted in

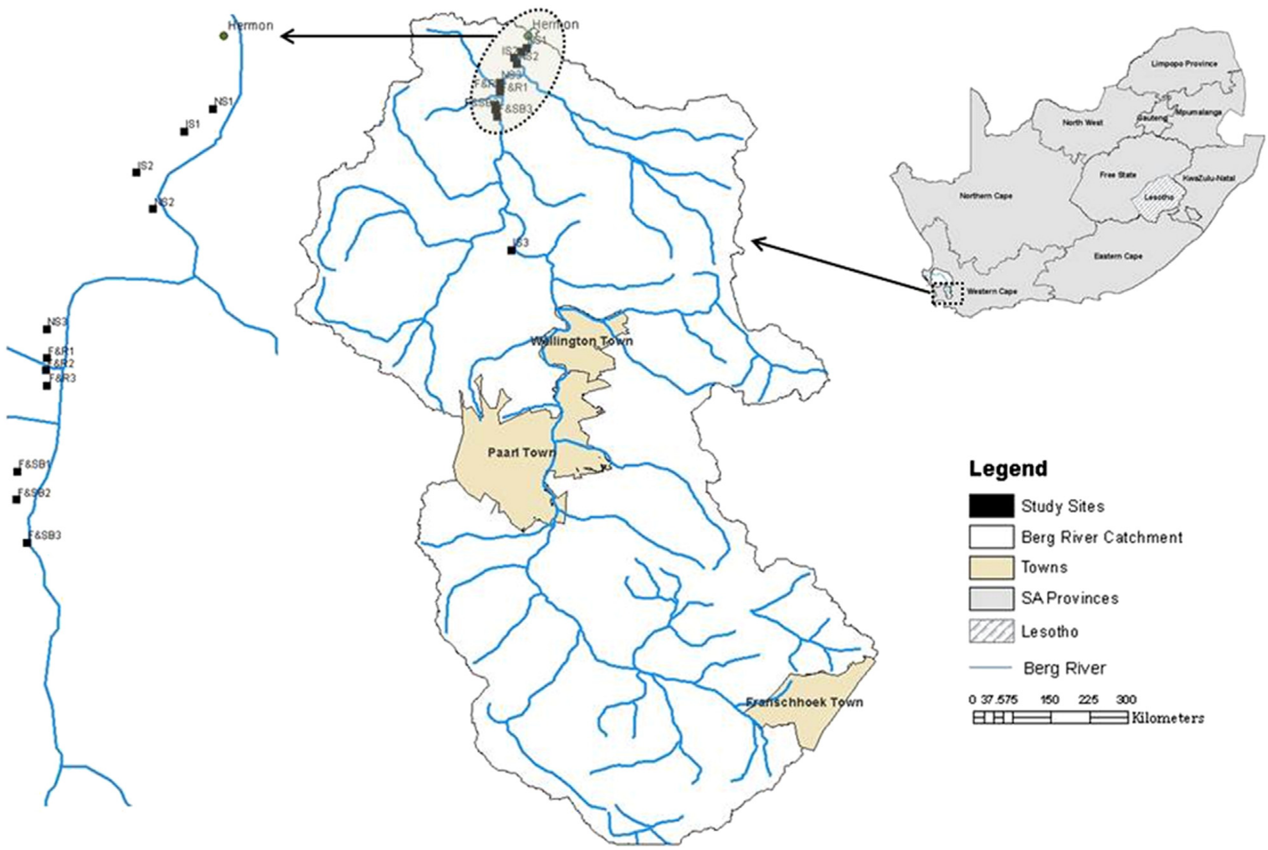

Fig. 1. Location of the study area and the different restoration sites namely fell & stack burning sites (F&SB), fell & removal sites (F&R), invaded sites (IS), and natural sites (NS), with each site replicated three times (e.g. IS 1, IS 2 and IS 3) in a restoration project along the Berg River in the Western Cape, South Africa.

March 2011. NS were dominated by native species and represented reference sites for restoring invaded sites. All sites were at least 200 m apart to provide a measure of independence (Galatowitsch and Richardson, 2005).

On F&R, F&SB and IS sites, twelve plots measuring 5 m × 5 m with a 5 m buffer zone were set up per site. Four of the 12 plots were used to assess natural recovery of species after alien clearing (passive restoration). The remaining eight plots were used for active restoration where success of seed broadcasting (on four plots) and of cuttings (on the other four plots) was tested. Only four plots were set-up in NS to determine presence of existing species. Corners of plots were permanently marked with metal fence droppers.

2.2.1. Selected restoration species

Nine native species, viz. *Diospyros glabra* (L.) De Winter, *S. angustifolia* L., *Searsia undulata* Jacq., *O. europaea* subsp. *africana* (Mill.) P.S. Green, *K. africana* (L.), *Euclea tomentosa* E. Meyer ex Drège, *M. major* (L.), *Metalasia muricata* (L.) D. Don and *Leonotis leonurus* (L.) R. Br, were broadcast sown and three cuttings of *D. glabra* (L.) De Winter, *O. europaea* subsp. *africana* (Mill.) P.S. Green and *Salix mucronata* subsp. *hirsuta* were planted in each of the active restoration plots (Table 1). These species were selected because they are local pioneers which recruit easily from seeds or cuttings (Holmes et al., 2008). They were also found along the Berg River, making the harvesting of large quantities of locally adapted seeds practical. Seeds and cuttings were collected from remnant individuals along the river from July 2010 until dehiscence and dispersal occurred, except for *M. muricata* and *L. leonurus* which were commercially sourced. Seed broadcasting was conducted in April 2011 (autumn) following the suggestion of Holmes et al. (2008) that seeding fynbos plants during this time and sowing a reasonably large quantity per plot enhances the chances of recruitment. Planting of cuttings was conducted in June 2011 (winter) when soils were wet due to winter rains, and a rooting hormone, Dynaroot B2, was used to facilitate root establishment. In an effort to address germination shortfalls, we adopted Doust et al.'s (2006) suggestion of burying our broadcast seeds with a layer of soil (approximately 5 mm). We used shallow seed burial even though it exposes seeds to a higher risk of predation compared to deeper burial because previous studies have shown that deeper burial can reduce seedling emergence (Christian and Stanton, 2004). No germination pre-treatment was administered on seeds sown in the field.

Table 1
Native species and seed quantities sown per plot in fell & stack burning, fell & removal and invaded sites along the Berg River, Western Cape, South Africa. Numeric counts are verified counts of the used broadcast quantity.

Species	Family	Seed broadcast quantities	Numeric estimates per plot	Numeric estimate per m^2
Harvested				
Diospyros glabra	Ebenaceae	One handful	150	6
Searsia angustifolia	Anacardiaceae	Two table spoon	150	6
Olea europaea sub africana	Oleaceae	One handful	120	4.8
Kiggelaria africana	Achariaceae	One handful	150	6
Melianthus major	Melianthaceae	One handful	150	6
Searsia undulata	Anacardiaceae	Two table spoon	150	6
Euclea tomentosa	Ebenaceae	One table spoon	60	2.8
Commercially sourced				
Metalasia muricata	Asteraceae	50 seeds[a]	50	2
Leonotis leonurus	Lamiaceae	50 seeds[a]	50	2

[a] Not measured but estimated at 50 seeds.

2.3. Germination pre-treatment experiment

Sixty soil samples, measuring approximately 28 cm wide × 30 cm long x 10 cm deep were excavated from NS along the Berg River. After excavation, the samples were placed into plastic trays of similar abovementioned dimension and transported to a passively ventilated greenhouse where air temperatures closely approximated outdoor conditions. The experimental layout comprised 6 tables (each table with ten trays) located at different positions in the greenhouse, with each table representing one of the six administered germination pre-treatments. At each table, five trays were sown with seven seeds of four species per tray, namely, *D. glabra*, *K. africana*, *L. leonurus* and *M. major*. The remaining five trays were sown with seven seeds each of *M. muricata*, *O. europaea*, *S. angustifolia*, *S. undulata* and *E. tomentosa*. Species had to be grouped this way as trays were too small to accommodate all species together and we wished to avoid the negative effects of seedling competition. Seeds were sown to a depth of 25 mm in autumn (April) 2011 and these were monitored weekly till early summer (late October) 2011. Trays were weeded weekly to remove non-target species. Water was supplied daily by an automated irrigation system over the entire experimental period (irrigating approximately 5 mm per day). Tables and trays were rotated monthly to account for minor variations in air temperature, light intensity and amounts of water dispensed within the greenhouse.

2.3.1. Germination pre-treatments

Prior to sowing, the following six germination treatments were carried out independently on the above mentioned six tables. On the first table a water soaking treatment was conducted. Water was boiled and poured into different heat resistant non-corrosive beakers containing the seeds. The seeds were left in the water for 24 h to allow the water to cool and the seeds to soak at room temperature. After 24 h the seeds were removed and drained before being sown into trays. Tiny seeds, particularly those of *L. leonurus* and *M. muricata*, were enclosed in sealed filter paper sachets before being soaked. On table two a heating treatment was conducted. Seeds were put in an oven and heated at 60 °C for 60 min. After heating they were allowed to cool at room temperature. The seed treatments were selected after consultation with experts (Anthony Hitchcock, SANBI, pers. comm., Sept 2010) to stimulate germination of hard-coated seeds. A smoking treatment was administered on table three. Seeds were first sown into germination trays and transferred to a smoking room, where a mixture of dry and green fynbos leaf and stem material was ignited and the smoke blown underneath the trays for approximately 2 h. Upon completion, the trays were transferred back to a greenhouse. Mechanical scarification was conducted on table four. Seed coats were pierced using a sharp knife. Tiny seeds of *L. leonurus* and *M. muricata* were lightly rubbed with the back of a knife to crack the seed coats. Seeds were then immediately sown in trays. Chemical scarification was conducted on table five. Seeds were put into heat resistant non-corrosive beakers and sulphuric acid (98% H_2SO_4) was added until all seeds were covered. The seeds were left for 15 min after which they were removed by thoroughly washing the acid off in water and drained off into another beaker. The seeds were then sown in germination trays. Lastly, no treatment was administered on table six as this acted as the control where seeds were sown into trays without any pre-treatment.

2.4. Data collection

On plots where natural recovery was monitored, detailed vegetation surveys were undertaken during spring of 2011 and summer of 2012. Spring was selected as it is the time during which most herbaceous species should be apparent, whereas summer was selected to assess contribution of typical dry conditions to restoration. Within each plot, total vegetation cover for both indigenous and alien plants (mostly herbaceous and graminoids) was estimated (to the nearest 5% or to the nearest 1% when species occupied <5%) as a percentage of the 1 m^2 quadrat placed at the edge of the plot and the entire plot (25 m^2). Species richness for all herbs and graminoids was determined from counts of the total numbers of individual plant species (indigenous and alien) present in a 1 m^2 quadrat, whilst species richness of trees and shrubs was measured in 25 m^2 plots. Species were also assigned to growth forms based on morphology and maximum height reached, as described by Goldblatt and Manning (2000). The four broad growth form classes used in this study are trees, shrubs, forbs (herbaceous plants) and graminoids. All recognisable species were collected in the field for identification. Species were labelled as native or alien following the criteria of Pyšek et al. (2004) and using published floras including Goldblatt and Manning (2000), Henderson (2001) and Bromilow (2010). Species which could not be positively identified were collected and labelled with a unique specimen number and sent to Compton Herbarium, South African National Biodiversity Institute (SANBI) for identification.

On plots where seeds and cuttings were sown and planted, recruitment success was monitored seasonally over one year (from winter 2011 to winter 2012). Monitoring included counting the total number of seeds that germinated and cuttings that established. Similarly, at the end of the greenhouse experiment, the number of seedlings that germinated from the different germination pre-treatments was counted and expressed as percentage of the total seeds sown.

2.5. Data analysis

After checking for normality using the Shapiro–Wilk and Kolmogorov–Smirnov test and proof of homogeneity of variance using Levene's test, the effects of the different passive and active restoration treatments on germination and vegetation variables (native and indigenous vegetation cover and indices of diversity (species richness, Shannon-Wiener, Simpson's index of diversity and evenness index)) were compared using one-way and repeated measures analysis of variances (ANOVA) as provided in STATISTICA version 10 (Statsoft Inc, 2010). The effects of the different germination pre-treatments on percentage germination in the greenhouse were compared using one-way ANOVA. Repeated measures ANOVA were used to determine changes between clearing treatments over seasons since clearing (winter, spring and summer). Where data were not normally distributed, arcsine transformations were applied. Where ANOVA's were significant, Tukey's HSD unequal *n* test was used to determine variance at $P < 0.05$. Statistical significance was determined at $P < 0.05$.

3. Results

3.1. Passive restoration

Species recovery after *E. camaldulensis* removal on fell and stack burning (F&SB) and fell and removal (F&R) was dominated by herbs and graminoids (Table 2), mostly alien herbs e.g. *Solanum nigrum*, *Rumex crispus* and *Lactuca serriola* and alien grasses (*Bromus catharticus* and *Avena fatua*) appearing in almost all F&R plots during spring (Appendix A). The recorded high frequencies of alien herbs and graminoids during spring translated into significantly ($P < 0.001$) higher cover of these two growth forms in F&R sites compared to invaded sites (IS) and natural sites (NS: Table 2). Both natives and aliens in their categorised growth forms showed significant differences ($P < 0.001$) amongst the different clearing treatments in both measured plot sizes (1 m^2 and 25 m^2: Table 2). However, there were no significant ($P > 0.05$) interactions between clearing treatments and seasons in alien trees and shrubs (in 1 m^2 plots) as well as in all natives (combined cover of all growth forms per 1 m^2 plots).

Species richness, Shannon-Wiener and Simpson's indices of diversity all differed significantly amongst the different clearing treatments and different seasons ($P < 0.001$: Fig. 2). The Tukey's test indicated that F&R had higher indices (species richness, Shannon-Wiener

Table 2
Species percentage cover recorded in different clearing treatments over two seasons in a restoration study along the Berg River in the Western Cape, South Africa.

Plant growth variable	Plot size	Spring 2011				Summer 2012				Repeated ANOVA F values, measures within subject effects		
		Fell & stack burning	Fell & removal sites	Invaded sites	Natural sites	Fell & stack burning	Fell & removal sites	Invaded sites	Natural sites	Clearing treatments	Season	Clearing treatments and seasons
Natives												
All natives	1 m^2	5.00 ± 1.07^{c}	28.75 ± 3.70^{b}	8.75 ± 1.25^{c}	54.58 ± 4.82^{a}	6.67 ± 0.94^{c}	32.50 ± 3.29^{b}	10.42 ± 1.14^{c}	49.58 ± 3.61^{a}	71.11***	0.15^{ns}	2.02^{ns}
	25 m^2	13.75 ± 2.14^{b}	49.58 ± 3.45^{a}	11.67 ± 2.91^{b}	60.42 ± 4.01^{a}	9.58 ± 1.56^{c}	37.50 ± 3.77^{b}	11.67 ± 2.91^{c}	65.83 ± 3.30^{a}	91.63***	2.80^{ns}	5.19*
Trees & shrubs	1 m^2	00	00	00	00	00	00	00	00	na	na	na
	25 m^2	0.00 ± 0.00^{c}	6.25 ± 1.64^{b}	9.58 ± 3.17^{b}	60.42 ± 4.01^{a}	0.00 ± 0.00^{b}	6.25 ± 1.64^{b}	9.58 ± 3.17^{b}	60.42 ± 4.01^{a}	107.7***	na	na
Herbs	1 m^2	2.92 ± 0.96^{c}	31.25 ± 3.70^{a}	3.33 ± 1.12^{c}	14.58 ± 3.28^{b}	0.42 ± 0.42^{a}	0.00 ± 0.00^{a}	0.00 ± 0.00^{a}	0.00 ± 0.00^{a}	26.18***	98.68***	26.58***
	25 m^2	11.25 ± 2.55^{b}	48.33 ± 3.50^{a}	6.67 ± 2.23^{b}	18.75 ± 4.31^{b}	0.83 ± 056^{b}	4.58 ± 0.42^{a}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	39.96***	149.23***	26.24***
Graminoids	1 m^2	0.00 ± 0.00^{b}	14.58 ± 1.89^{a}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	00	00	00	00	na	na	na
	25 m^2	0.00 ± 0.00^{b}	35.83 ± 5.96^{a}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	00	00	00	00	na	na	na
Aliens												
All aliens	1 m^2	37.92 ± 4.19^{b}	59.58 ± 3.04^{a}	17.50 ± 2.85^{c}	6.25 ± 1.64^{c}	25.83 ± 5.90^{a}	22.08 ± 3.04^{a}	18.33 ± 2.64^{a}	4.17 ± 1.04^{b}	40.03***	31.69***	14.90***
	25 m^2	49.58 ± 5.85^{b}	65.83 ± 2.37^{a}	58.75 ± 2.83^{ab}	10.83 ± 2.81^{c}	41.67 ± 6.61^{b}	32.00 ± 4.75^{b}	58.75 ± 2.83^{a}	7.08 ± 2.42^{c}	48.41***	18.04***	8.18***
Trees & shrubs	1 m^2	12.92 ± 4.82^{a}	8.33 ± 2.07^{ab}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	4.17 ± 1.20^{a}	3.75 ± 1.52^{a}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	9.45***	5.75*	2.29^{ns}
	25 m^2	24.58 ± 7.96^{bc}	33.33 ± 3.81^{b}	56.25 ± 3.15^{a}	7.08 ± 3.23^{c}	8.33 ± 1.12^{c}	22.08 ± 3.56^{b}	56.25 ± 3.15^{a}	7.08 ± 13.23^{c}	37.41***	10.41**	3.81*
Herbs	1 m^2	31.25 ± 5.40^{b}	46.25 ± 2.83^{a}	3.75 ± 1.25^{c}	3.75 ± 1.86^{c}	29.17 ± 6.33^{a}	21.25 ± 3.60^{a}	2.50 ± 0.75^{b}	3.33 ± 1.12^{b}	40.35***	9.74***	6.67***
	25 m^2	47.92 ± 6.47^{b}	64.17 ± 2.53^{a}	4.17 ± 1.49^{c}	6.67 ± 2.41^{c}	38.75 ± 7.39^{a}	30.33 ± 5.24^{a}	3.75 ± 1.09^{b}	2.92 ± 0.96^{b}	56.87***	18.08***	7.44***
Graminoids	1 m^2	5.83 ± 1.93^{b}	17.50 ± 3.67^{a}	2.50 ± 0.97^{b}	4.17 ± 1.20^{b}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	1.67 ± 0.71^{a}	0.00 ± 0.00^{b}	8.42***	39.25***	10.28***
	25 m^2	7.08 ± 2.85^{b}	31.25 ± 2.05^{a}	6.25 ± 2.55^{b}	10.00 ± 3.08^{b}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	4.58 ± 1.89^{a}	0.00 ± 0.00^{b}	14.86***	81.57***	21.95***

Data are means ± se and results of repeated measures ANOVAs are shown ($^{*}P < 0.05$, $^{**}P < 0.01$, $^{***}P < 0.001$).
Values within columns with different letter superscripts are significantly different.
NS = not significant.
na = no statistical comparisons could be done.

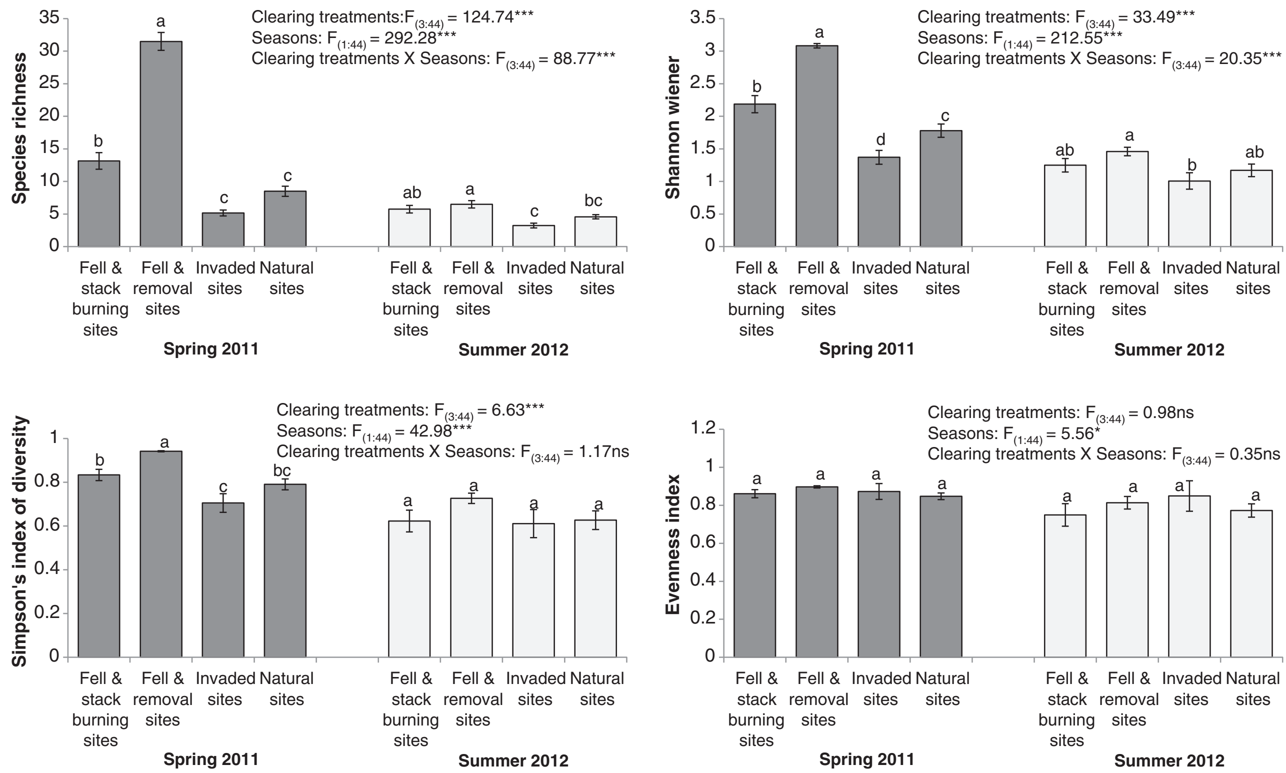

Fig. 2. Indices of diversity in different clearing treatments, namely fell & stack burning (F&SB), fell & removal (F&R), invaded (IS) and natural sites (NS) over two seasons along the Berg River in the Western Cape, South Africa. Bars are means ± se and results of repeated measures ANOVAs are shown (*P < 0.05, **P < 0.01, ***P < 0.001). Bars with different letter superscripts are significantly different. NS = not significant; *P > 0.05.

and Simpson's index of diversity) than all the other treatments. All indices of diversity were low in summer compared to those in spring and interactions between clearing treatments and seasons were significantly different for species richness and Shannon-Wiener ($P < 0.001$) but not for Simpson's index of diversity and evenness ($P > 0.05$: Fig. 2).

3.2. Active restoration

Germination differed amongst clearing treatments and seasons (Table 3). With the exception of *S. undulata* that did not germinate in any treatments in all seasons and *E. tomentosa* as well as *K. africana*, whose germination rates showed no significant differences ($P > 0.05$) amongst the different clearing treatments, all other species showed significantly different germination rates amongst the different clearing treatments ($P < 0.05$). Highest germination rates for all the species were recorded in F&SB compared to F&R and IS in all seasons. Seasonality comparisons show significantly different germination rates amongst the different seasons ($P < 0.001$; Table 3). However, significant ($P < 0.001$) interactions between clearing treatments and seasons were only apparent in *D. glabra*, *S. angustifolia*, *M. major* and *L. leonurus* (although no *L. leonurus* seeds germinated in invaded sites).

Seedling survival after the summer drought was low due to the recorded high mortality rate for all species amongst the different clearing treatments (Fig. 3). Species showed no significant differences in mortality rates amongst the different clearing treatments ($P < 0.05$). *M. muricata* and *K. africana* showed high mortality rates in F&SB (95% and 91% respectively) whereas *O. europaea* showed low mortality rates (65%) in the same clearing treatment. In F&R, *K. africana* and *L. leonurus* had the highest mortality rate of 94% and 93% respectively compared to *M. muricata* for which the lowest mortality rate of 56% was recorded in the same clearing treatment. In IS, only for *S. angustifolia* a low mortality rate of 50% was recorded, with all other species having mortality rates of more than 80% (Fig. 3).

Cuttings of the three targeted restoration species failed to establish in all treatments by the end of spring, so no statistical analyses could be done. Some cuttings of *S. mucronata* developed green leaves by the end of winter, but all had died by the end of spring.

3.3. Germination pre-treatments

With the exception of *O. europaea* and *E. tomentosa* which showed no significant differences ($P > 0.05$) amongst the different germination pre-treatments, all other species showed significantly different germination rates amongst the different clearing treatments ($P < 0.001$: Table 4). For *D. glabra*, *O. europaea* and *L. leonurus* highest germination rates were recorded in control treatments, whereas, *M. major* and *E. tomentosa* showed high germination rates after heating treatment and *S. angustifolia* after mechanical scarification (Table 4). *M. muricata* only germinated after a smoke treatment (46%), whereas *K. africana*, which experienced the lowest germination rates in all pre-treatments, had its highest germination in chemical scarification (26%).

4. Discussion

The broader objective of this study was to compare the effectiveness of passive and active restoration in promoting native riparian species recovery following two clearing treatments: fell & remove and fell & stack burn. Our results indicate that both passive and active restoration following the two clearing treatments faced several challenges. There was no recruitment of native species in our passive restoration site. Previous work on passive restoration in the Western Cape has shown mixed results. Some studies have shown good recovery where previous invasion densities were low (Reinecke et al., 2008; Galatowitsch and

Richardson, 2005) whereas others have shown poor recovery where invasion densities were high (Blanchard and Holmes, 2008). Recruitment of native species following active restoration was affected by low germination of introduced species. Furthermore, the few seeds that germinated in both clearing treatments were affected by high seedling mortality rate in summer and competition from alien herbs and graminoids (secondary invasion), thus making native species recovery a challenge. Our active restoration results are in contrast with those of Pretorius et al. (2008) who, 8 years after the initial sowing treatments on riparian systems at Oaklands farm in the Western Cape, reported the presence of few native species on restoration sites. Clearly, many factors interact to influence recruitment success at any site. Besides, we are presenting results from only one year after the initial clearing. Preliminary results are crucial for setting the direction of the recovery succession and are fundamental to the development of evidence-based restoration solutions.

4.1. Passive restoration

Our assessment of natural recovery on cleared sites showed a complete absence of seedling recruitment. The most notable feature of the vegetation at our cleared sites was the high cover of alien herbs and graminoids. The proliferation of alien herbs and graminoids after alien clearing has been reported in the past (Holmes and Richardson, 1999; Richardson et al., 2000; Yelenik et al., 2004); their dominance has been attributed to soil nutrient enrichment, a legacy effect from prior invasion (Yelenik et al., 2004). Balamurungan et al. (2000) showed that soils beneath *Eucalyptus* stands have increased soil nutrients mainly due to abundant decayed litter produced by the plant. We therefore suspect that soils at our site had increased nutrient levels after alien removal which stimulated the growth of alien herbaceous species and graminoids. Competition by alien species has been shown to negatively affect the growth of native seedlings (D'Antonio and Mack, 2001). Furthermore, studies have shown that alien herbs tend to use large amounts of water, thereby limiting water supply needed for survival of woody native plant seedlings (Rey Benayas et al., 2007).

Fire stimulates germination of *Acacia* seeds (Richardson and Kluge, 2008). Although not abundant above ground prior to restoration treatments, germination of *A. mearnsii* was high in both F&R and F&SB — the result of the presence of *A. mearnsii* seeds in the soil-stored seed bank. We observed that growth of *A. mearnsii* and alien herbaceous species and graminoids in F&SB mainly occurred on the edges of plots. This could be because fire intensity at the centre of plots was high enough to kill seeds whereas the lower temperatures and heat duration on the edges were more conducive to breaking dormancy and stimulating germination. Furthermore, very intense fires induce soil water repellency which reduces seed germination and seedling survival since water infiltration is reduced (Scott et al., 1998; Ruwanza et al., 2013b).

One reason for a lack of native species germination could be a depleted native soil seed bank (Hobbs and Harris, 2001; Holmes et al., 2008). Several studies have confirmed that native soil seed banks do become depleted after several decades of invasion by alien trees (Holmes et al., 2008; Vosse et al., 2008). On the other hand, some seeds remain dormant for many years (Velempini et al., 2003). Therefore, monitoring and follow-up efforts should be conducted for several years. In this study, the monitoring process will include years of assessing the effectiveness of both passive and active restoration, implying that the final restoration conclusion will be drawn after decades.

Interestingly, we observed the presence of established native trees and shrubs (shade-tolerant species that were present prior to clearing) in F&R sites. These include *D. glabra*, *M. major*, *K. africana* and *S. angustifolia*. The presence of these species presents opportunities for recovery initiating from these remnant foci (Guevara et al., 1986; Galatowitsch and Richardson, 2005). Their presence also facilitates the establishment of other native plants by ameliorating the existing harsh microclimatic conditions associated with *E. camaldulensis* cover. They also assist by outcompeting recruiting alien herbs and graminoids for resources (nutrients and water) thereby reducing growth and establishment of these secondary invaders (Duncan and Chapman, 1999).

4.2. Active restoration

It is important to test for seed viability at the onset of any active restoration experiment (Holmes et al., 2008). Although our greenhouse experiment showed that harvested seeds were viable (germination above 50% especially in control treatments), recruitment on active restoration plots (germination below 30%) was generally low across treatments and seasons.

The poor germination rates recorded under field conditions could be due to several environmental and seedbed (soil) factors (Battaglia et al., 2000). It is difficult to pinpoint the exact factors that prevented germination in our field experiment as these were not tested. However, we

Table 3
Germination percentages calculated from seedling counts done in winter (2011), spring (2011), summer (2012) and winter (2012) of nine target native species broadcasted into three restoration treatments.

	Winter 2011			Spring 2011		
Species/treatments	Fell & stack burning	Fell & removal sites	Invaded sites	Fell & stack burning	Fell & removal sites	Invaded sites
Harvested seeds						
Diospyros glabra	12.80 ± 3.35^{a}	7.11 ± 2.35^{ab}	3.17 ± 1.53^{b}	16.83 ± 3.77^{a}	11.33 ± 2.68^{a}	10.28 ± 1.97^{a}
Searsia angustifolia	45.72 ± 5.63^{a}	8.78 ± 2.78^{b}	1.44 ± 0.59^{b}	55.44 ± 5.49^{a}	12.00 ± 3.49^{b}	2.72 ± 1.26^{b}
Olea europaea sub africana	2.92 ± 0.66^{a}	0.00 ± 0.00^{b}	1.53 ± 0.71^{ab}	2.78 ± 0.77^{a}	0.00 ± 0.00^{c}	1.67 ± 0.00^{ab}
Kiggelaria africana	12.11 ± 2.00^{a}	7.06 ± 1.84^{a}	8.28 ± 1.89^{a}	14.94 ± 2.19^{a}	9.06 ± 2.43^{a}	11.50 ± 3.00^{a}
Melianthus major	45.67 ± 3.68^{a}	22.44 ± 2.20^{b}	3.78 ± 1.78^{c}	51.89 ± 3.84^{a}	26.94 ± 2.03^{b}	4.67 ± 1.98^{c}
Searsia undulata	00	00	00	00	00	00
Euclea tomentosa	16.25 ± 3.65^{a}	14.31 ± 5.13^{a}	15.28 ± 3.55^{a}	22.78 ± 4.08^{a}	19.58 ± 6.67^{a}	20.00 ± 4.22^{a}
Commercially sourced seeds						
Metalasia muricata	15.67 ± 5.99^{a}	13.50 ± 4.97^{ab}	0.00 ± 0.00^{b}	25.00 ± 12.18^{a}	15.67 ± 5.69^{a}	0.00 ± 0.00^{a}
Leonotis leonurus	41.17 ± 7.52^{a}	18.33 ± 5.72^{b}	0.00 ± 0.00^{b}	51.00 ± 9.08^{a}	21.67 ± 6.10^{b}	0.00 ± 0.00^{b}

Data are mean ± se and results of repeated measures ANOVAs are shown (*P < 0.05, **P < 0.01, ***P < 0.001).
Values within columns with the different letter superscripts are significantly different.
NS = not significant.
na = no statistical comparisons could be done.

assume that temperature could have been important. Most of our seeds are known to germinate best under relatively hot day temperatures and cool nights, which allows the testas to crack, thus permitting water to enter and initiate germination (Anthony Hitchcock, SANBI, pers. comm., Sept. 2010). We broadcasted our seeds in autumn (April 2011) as suggested by Holmes et al. (2008) but we suspect that the above-average temperatures experienced were not conducive to breaking dormancy. However, we were surprised by the low germination rates in spring and summer. The low germination in summer could be because of the lack of water and subsequent low soil moisture levels associated with the dry summer, whereas in spring the recorded high cover of alien herbs and graminoids especially in F&R sites could have resulted in intense competition for soil moisture and light (Reinecke et al., 2008; Yelenik et al., 2004) which could have suppressed native species germination. Furthermore, our experiment was conducted five months after clearing and the observed *Eucalyptus* litter layer could have provided a physical barrier to germination (Facelli et al., 1999).

During seed broadcasting we buried our seeds at a depth of about 5 mm following the recommendation of Doust et al. (2006) to enhance germination, but we recorded low germination rates. Seed burial could have negatively affected seed germination, possibly by causing seed death (due to pathogens) before germination, predation or persistence in a dormant state (Burmeier et al., 2010). We observed rodents on the sites so they could have predated the seeds — Bond and Breytenbach (1985) showed that seed predation by rodents is generally high in the fynbos.

The high mortality rate of our seedlings could be due to the high temperatures and low rainfall normally experienced in summer. High temperatures affect seedling growth by increasing evaporative demand and direct tissue damage where seedlings are in contact with hot soil surfaces (Kolb and Robberecht, 1996). The lack of water during summer is also associated with seedling transpiration water loss which is mainly induced by high soil surface temperatures.

4.3. Recommendations for passive restoration

The few identified remnant native species present within F&R should be protected from accidental clearing and damage from herbicide over-spraying during follow-up operations to remove emerging aliens. Once the alien trees are felled, removal presents better results by minimising remnant species damage compared to stack burning which killed both the existing remnants and the soil-stored seed bank. In this regard, F&R seems to be the most appropriate method for facilitating recovery of remnant native species. However, according to Holmes et al. (2008) F&R is more applicable where key biotic and abiotic thresholds have not been crossed (e.g. seed bank not severely depleted) i.e. the native ecosystem is still resilient, which is more likely on sites not heavily invaded or degraded.

Clearing alone resulted in the establishment of a system dominated by alien herbaceous species and graminoids. If the key factor precipitating the dominance of these alien weeds is the high soil-nutrient levels associated with invaded areas (Yelenik et al., 2004), then we suggest that combining active and passive restoration mechanisms to reduce soil nutrient levels should be applied. Such soil-nutrient reduction mechanisms include C and Ca addition to reduce soil N and P levels or soil transfer. However, such methods are probably unrealistic for large scale restoration as they are extremely labour intensive and expensive. Other methods that can be used to reduce dominance of alien weeds on cleared sites targeted for passive restoration include spraying weeds with herbicides as a follow-up treatment one year after clearing. Also, thinning of the invasive species instead of complete clearing on sites targeted for passive restoration has been shown to reduce the dominance of alien herbaceous species and graminoids (Ruwanza et al., 2013a). It also creates perches of recovering native understory vegetation that can be used by birds to disperse seeds of native species (Heelemann et al., 2012).

4.4. Recommendations for active restoration

The relatively high germination of three species (*L. leonurus*, *M. major* and *S. angustifolia*) in F&SB, mainly due to reduced alien herbaceous species and graminoids competition, suggests that F&SB facilitates species germination better than F&R. However, the high mortality rates in both F&SB and F&R sites recorded during summer points to the limited role played by both seed broadcasting and planting of cuttings in the establishment of native species following alien removal. Several environmental and soil related constraints seem to affect germination and seedling establishment. To overcome some of the environmental and soil-related constraints we suggest seeding native species during the appropriate season. Appropriate seeding seasons are species dependent although Holmes et al. (2008) recommend seeding in autumn for most fynbos and renosterveld species. Also, barriers to seed penetration after broadcasting e.g. leaf litter from the previous invader and hard soil crust should be minimised by removing the litter layer as well as sowing when soil surface is moist.

Table 3 (*continued*)

Summer 2012			Winter 2012			Repeated ANOVA F values, measures within subject effects		
Fell & stack burning	Fell & removal sites	Invaded sites	Fell & stack burning	Fell & removal sites	Invaded sites	Clearing treatments	Seasons	Clearing treatments and seasons
16.44 ± 2.97^{a}	5.06 ± 1.18^{b}	0.78 ± 0.62^{b}	2.72 ± 0.64^{a}	1.56 ± 0.56^{ab}	0.56 ± 0.21^{b}	9.22^{***}	18.33^{***}	2.57^{*}
38.94 ± 5.17^{a}	4.28 ± 0.89^{b}	1.11 ± 0.63^{b}	6.17 ± 1.57^{a}	1.44 ± 0.36^{b}	0.28 ± 0.13^{b}	76.88^{***}	36.81^{***}	18.89^{***}
0.69 ± 0.69^{a}	0.00 ± 0.00^{a}	0.76 ± 0.43^{a}	0.90 ± 0.33^{a}	0.00 ± 0.00^{b}	0.28 ± 0.16^{ab}	10.07^{***}	5.94^{***}	2.04^{ns}
4.06 ± 0.57^{a}	3.89 ± 0.97^{a}	6.06 ± 1.20^{a}	1.33 ± 0.25^{a}	0.50 ± 0.12^{b}	0.50 ± 0.20^{b}	1.4^{2ns}	43.81^{***}	1.73^{ns}
24.00 ± 2.43^{a}	10.68 ± 2.32^{b}	2.00 ± 1.32^{c}	6.56 ± 1.34^{a}	3.06 ± 0.77^{b}	0.11 ± 0.07^{b}	87.12^{***}	113.49^{***}	26.51^{***}
00	00	00	00	00	00	na	na	na
13.19 ± 1.85^{a}	3.89 ± 0.97^{a}	6.06 ± 1.20^{a}	2.92 ± 1.14^{a}	1.81 ± 0.56^{a}	3.33 ± 1.26^{a}	0.38^{ns}	24.48^{***}	0.36^{ns}
10.83 ± 3.68^{a}	4.17 ± 3.66^{ab}	0.00 ± 0.00^{b}	2.00 ± 1.15^{a}	3.00 ± 2.29^{a}	0.00 ± 0.00^{a}	3.69^{*}	5.98^{***}	1.88^{ns}
14.33 ± 3.66^{a}	3.67 ± 2.19^{b}	0.00 ± 0.00^{b}	5.00 ± 1.90^{a}	1.50 ± 1.01^{ab}	0.00 ± 0.00^{b}	22.19^{***}	25.25^{***}	8.84^{***}

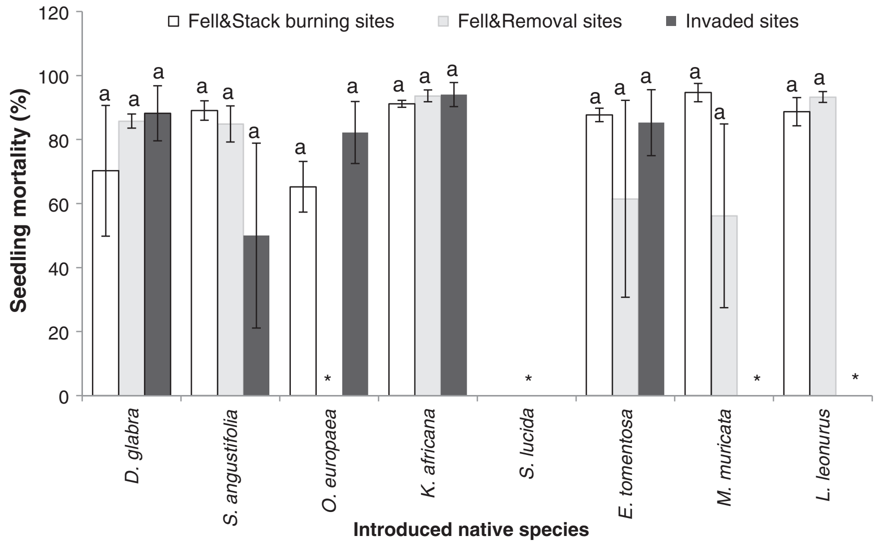

Fig. 3. Mortality (%) of nine sown native species in different clearing treatments, namely fell & stack burning (F&SB), fell & removal (F&R), and invaded (IS) along the Berg River in the Western Cape, South Africa. Bars are means ± se and bars with different letter superscripts are significantly different. (*) indicates no germination thus no mortality.

Where fire is used to kill invasives, the study suggests that mechanisms that limit water repellency of soils, improve soil-related ecosystem functions, soil chemistry and soil physical properties to facilitate restoring indigenous vegetation composition, structure and species richness should be applied (Ruwanza et al., 2013b). Such mechanisms include tilling cleared sites (Hallett, 2007), applying soil surfactants on cleared sites either as liquid through irrigation or as granular material (Moore et al., 2010) or overlaying cleared sites with a clay-rich soil layer (Wallis and Horne, 1992).

Selection of appropriate species that are likely to germinate is important for effective active restoration. Local seeds, sourced from along the same river or close to the riparian system being restored, should be used to avoid genetic contamination (Broadhurst et al., 2008). Furthermore, priority should be given to species that germinate rapidly during brief periods of favourable conditions without any pre-treatments and also to species that have the potential to germinate and survive under dry and harsh conditions. Characteristics of species that adapt to dry conditions include the ability to develop deep tap roots that allow acquisition of underground water in summer. Morphological and physiological characteristics of such species include a high leaf area to stem diameter ratio which allows effective stem cooling during heat and the ability to maintain a high stomatal conductance at high temperature which promotes transpirational heat dissipation (Kolb and Robberecht, 1996).

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.sajb.2013.06.022.

Table 4
Effects of different germination pre-treatments on nine target native species tested under greenhouse conditions.

	Soaking	Heating	Smoking	Mechanical scarification	Chemical scarification	Control	ANOVA = F(5;24)
Harvested seeds							
Diospyros glabra	5.71 ± 3.50^{d}	85.71 ± 6.39^{ab}	68.57 ± 11.43^{b}	97.14 ± 2.86^{a}	34.29 ± 11.61^{c}	100.0 ± 7.40^{a}	26.3^{***}
Searsia angustifolia	68.57 ± 9.48^{ab}	74.29 ± 9.48^{ab}	17.14 ± 5.35^{c}	88.57 ± 11.43^{a}	54.29 ± 12.29^{b}	77.14 ± 9.69^{ab}	6.5^{***}
Olea europaea subsp. *africana*	65.71 ± 13.25^{a}	54.29 ± 5.35^{a}	45.71 ± 12.29^{a}	51.43 ± 17.84^{a}	60.00 ± 16.54^{a}	71.43 ± 9.04^{a}	0.5^{ns}
Kiggelaria africana	2.86 ± 2.25^{b}	8.50 ± 3.50^{b}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	25.71 ± 5.35^{a}	8.57 ± 3.50^{b}	9.2^{***}
Melianthus major	37.14 ± 7.28^{b}	97.14 ± 2.86^{a}	40.00 ± 15.25^{b}	20.00 ± 10.69^{b}	74.29 ± 9.48^{a}	74.29 ± 10.50^{a}	8.4^{***}
Searsia undulata	00	00	00	00	00	00	na
Euclea tomentosa	65.71 ± 9.69^{a}	80.00 ± 9.69^{a}	25.71 ± 12.25^{b}	54.29 ± 12.29^{ab}	54.29 ± 13.85^{ab}	54.29 ± 12.30^{ab}	2.3^{ns}
Commercially sourced seeds							
Metalasia muricata	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	45.71 ± 13.85^{a}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	0.00 ± 0.00^{b}	10.9^{***}
Leonotis leonurus	0.00 ± 0.00^{c}	94.29 ± 3.50^{ab}	85.71 ± 4.52^{b}	91.43 ± 3.50^{ab}	0.00 ± 0.00^{c}	97.14 ± 2.86^{a}	257.6^{***}

Data are means ± se and results of one-way ANOVAs are shown (*P < 0.05, **P < 0.01, ***P < 0.001).
Values within columns with the different letter superscripts are significantly different.
NS = not significant.
na = no statistical comparisons could be done.

References

Balamurungan, J., Kumaraswamy, K., Rajarajan, A., 2000. Effect of *Eucalyptus citriodora* on the physical and chemical properties of soils. Journal of the Indian Society of Soil Science 48, 491–495.

Battaglia, L.L., Foré, S.A., Sharitz, R.R., 2000. Seedling emergence, survival and size in relation to light and water availability in two bottomland hardwood species. Journal of Ecology 88, 1041–1050.

Blanchard, R., Holmes, P.M., 2008. Riparian vegetation recovery after invasive alien tree clearance in the fynbos biome. South African Journal of Botany 74, 421–431.

Bond, W.J., Breytenbach, G.J., 1985. Ants, rodents and seed predation in Proteaceae. South African Journal of Zoology 20, 150–154.

Broadhurst, L.M., Lowe, A., Coates, D.J., Cunningham, S.A., McDonald, M., Vesk, P.A., Yates, C., 2008. Seed supply for broadscale restoration: maximizing evolutionary potential. Evolutionary Applications 1, 587–597.

Bromilow, C., 2010. Problem Plants and Alien Weeds of South Africa. Briza Publications, Pretoria, South Africa.

Brown, N.A.C., Botha, P.A., 2004. Smoke seed germination studies and a guide to seed propagation of plants from the major families of Cape Floristic Region, South Africa. South African Journal of Botany 70, 559–581.

Budy, J., Evans, R., Young, J., 1986. Understanding Seed Handling for Germination. Volunteers in Technical Assistance, Arlington, VA.

Burmeier, S., Eckstein, R.L., Otte, A., Donath, T.W., 2010. Desiccation cracks act as natural seed traps in flood-meadow systems. Plant and Soil 333, 351–364.

Christian, C.E., Stanton, M.L., 2004. Cryptic consequences of a dispersal mutualism: seed burial, elaiosome removal, and seed-bank dynamics. Ecology 85, 1101–1110.

Crawley, M.J., 1992. Seed predators and plant population dynamics. In: Fenner, M. (Ed.), Seeds: The Ecology of Regeneration in Plant Communities. CAB, Wallingford, UK, pp. 157–192.

D'Antonio, C.M., Mack, M., 2001. Exotic grasses potentially slow invasion of an N-fixing tree into a Hawaiian Woodland. Biological Invasions 3, 69–73.

de Villiers, S., 2007. The deteriorating nutrient status of the Berg River, South Africa. Water SA 33, 1–6.

Doust, S.J., Erskine, P.D., Lamb, D., 2006. Direct seedling to restore rainforest species: microsite effects on the early establishment and growth of rainforest tree seedlings on degraded land in the wet tropics of Australia. Forest Ecology and Management 234, 333–343.

Doust, S.J., Erskine, P.D., Lamb, D., 2008. Restoring rainforest species by direct seeding: tree seedling establishment and growth performance on degraded land in the wet tropics of Australia. Forest Ecology and Management 256, 1178–1188.

Duncan, R.S., Chapman, C.A., 1999. Seed dispersal and potential forest succession in abandoned agriculture in Tropical Africa. Ecological Applications 9, 998–1008.

Dye, P.J., Poulter, A.G., 1995. A field demonstration of the effect on stream flow of clearing invasive pine and wattle trees from a riparian zone. South African Forestry Journal 173, 27–30.

Esler, K.J., Holmes, P.M., Richardson, D.M., Witkowski, E.T.F., 2008. Riparian vegetation management in landscapes invaded by alien plants: insights from South Africa. South African Journal of Botany 74, 401–552.

Facelli, J.M., Williams, R., Fricker, S., Ladd, B., 1999. Establishment and growth of seedlings of *Eucalyptus obliqua*: interactive effects of litter, water, and pathogens. Australian Journal of Ecology 24, 484–494.

Florentine, S.K., Graz, F.P., Ambrose, G., O'Brien,, L., 2011. The current status of different age, direct-seeded revegetation sites in an agricultural landscape in the Burrumbeet Region, Victoria, Australia. Land Degradation and Development.

Forsyth, G.G., Richardson, D.M., Brown, P.J., Van Wilgen, B.W., 2004. A rapid assessment of the invasive status of *Eucalyptus* species in two South African provinces. South African Journal of Science 100, 75–77.

Gaertner, M., Holmes, P.M., Richardson, D.M., 2012. Biological invasions, resilience and restoration. In: Van Andel, J., Aronson, J. (Eds.), Restoration Ecology: The New Frontier, second edition. Wiley-Blackwell, Oxford, pp. 265–280.

Galatowitsch, S., Richardson, D.M., 2005. Riparian scrub recovery after clearing of invasive alien trees in headwater streams of the Western Cape, South Africa. Biological Conservation 122, 509–521.

Geldenhuys, C.J., 2008. Practical guidelines for the rehabilitation of forest-related streambank vegetation with removal of invader plant stands along the Berg River, Western Cape. Working for Water Report (FW — 02/08), South Africa.

Goldblatt, P., Manning, J., 2000. Cape Plants. A Conspectus of the Cape Flora of South Africa. Strelitzia 9, National Botanical Institute, Pretoria.

Guevara, S., Purata, S.E., Van der Maarel, E., 1986. The role of remnant forest trees in tropical secondary succession. Plant Ecology 66, 77–84.

Hallett, P.D., 2007. An introduction to soil water repellency. Proceedings of the 8th International Symposium on Adjuvants for Agrochemicals (ISAA2007), Columbus, OH. ISBN: 978-0-473-12388-8.

Heelemann, S., Krug, C.B., Esler, K.J., Reisch, C., Poschlod, P., 2012. Pioneers and perches-promising restoration methods for degraded Renosterveld habitats? Restoration Ecology 20, 18–23.

Henderson, L., 2001. Alien weeds and invasive plants: a complete guide to declared weeds and invaders in South Africa. ARC-PPRI, PPRI Handbook No. 12, Pretoria, South Africa.

Hobbs, R.J., Harris, J.A., 2001. Restoration ecology: repairing the Earth's ecosystems in the new millennium. Restoration Ecology 9, 236–246.

Holl, K.D., Aide, T.M., 2011. When and where to actively restore ecosystems? Forest Ecology and Management 261, 1588-1563.

Holmes, P.M., Richardson, D.M., 1999. Protocols for restoration based on recruitment dynamics, community structure and ecosystem function: perspectives from South African fynbos. Restoration Ecology 7, 215–230.

Holmes, P.M., Esler, K.J., Richardson, D.M., Witkowski, E.T.F., 2008. Guidelines for improved management of riparian zones invaded by alien plants in South Africa. South African Journal of Botany 74, 538–552.

Hood, W.G., Naiman, R.J., 2000. Vulnerability of riparian zones to invasion by exotic vascular plants. Plant Ecology 148, 105–114.

Iponga, D.M., Krug, C.B., Milton, S.J., 2005. Competition and herbivory influence growth and survival of shrubs on old fields: implications for restoration of Renosterveld shrubland. Journal of Vegetation Science 16, 685–692.

Kolb, P.F., Robberecht, R., 1996. High temperature and drought stress effects on survival of *Pinus ponderosa* seedlings. Tree Physiology 16, 665–672.

Le Maitre, D.C., Gaertner, M., Marchante, E., Ens, E., Holmes, P.M., Pauchard, A., O'Farrell, P.J., Rogers, A.M., Blanchard, R., Blignaut, J., Richardson, D.M., 2011. Impacts of invasive Australian acacias: implications for management and restoration. Diversity and Distributions 17, 1015–1029.

Meek, C.S., Richardson, D.M., Mucina, L., 2010. A river runs through it: land use and the composition of vegetation along a riparian corridor in the Cape Floristic Region, South Africa. Biological Conservation 143, 156–164.

Meek, C.S., Richardson, D.M., Mucina, L., 2013. Plant communities along the Eerste River, Western Cape, South Africa: Community descriptions and implications for restoration. Koedoe 55 (1), 14. http://dx.doi.org/10.4102/koedoe.v55i1.1099.

Milton, S.J., 1995. Effects of rain, sheep and tephritid flies on seed production of two arid Karoo shrubs in South Africa. Journal of Applied Ecology 32, 137–144.

Moore, D., Kostka, S.J., Boerth, T.J., Franklin, M., Ritsema, C.J., Dekker, L.W., Oostindie, K., Stoof, C., Wesseling, J., 2010. The effect of soil surfactants on soil hydrological behavior, the plant growth environment, irrigation efficiency and water conservation. Journal of Hydrology and Hydromechanics 58, 142–148.

Mucina, L., Rutherford, M.C., 2006. The vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19.South African National Biodiversity institute, Pretoria.

Pretorius, M., Esler, K.J., Holmes, P.M., Prins, N., 2008. The effectiveness of active restoration following alien clearance in fynbos riparian zones and resilience of treatments to fire. South African Journal of Botany 74, 517–525.

Prinsloo, F.W., Scott, D.F., 1999. Stream flow responses to the clearing of invasive alien trees from riparian zones from three sites in the Western Cape Province. South African Forestry Journal 185, 1–7.

Pyšek, P., Richardson, D.M., Rejmánek, M., Webster, G.L., Williamson, M., Kirschner, J., 2004. Alien plants in checklists and floras: towards better communication between taxonomists and ecologists. Taxon 53, 131–143.

Reinecke, M.K., Pigot, A., King, J.M., 2008. Spontaneous succession of riparian fynbos: is unassisted recovery a viable restoration strategy? South African Journal of Botany 74, 412–420.

Rey Benayas, J.M., Fernández, A., Aubenau, A., 2007. Clipping herbaceous vegetation improves early performance of planted seedlings of the Mediterranean shrub *Quercus coccifera*. Web Ecology 7, 120–131.

Richardson, D.M., Kluge, R.L., 2008. Seed banks of invasive Australian *Acacia* species in South Africa: role in invasiveness and options for management. Perspectives in Plant Ecology, Evolution and Systematics 10, 161–177.

Richardson, D.M., Van Wilgen, B.W., 2004. Invasive alien plants in South Africa: how well do we understand the ecological impacts? South African Journal of Science 100, 45–52.

Richardson, D.M., Macdonald, I.A.W., Hoffmann, J.H., Henderson, L., 1997. Alien plant invasion. In: Cowling, R.M., Richardson, D.M., Pierce, S.M. (Eds.), Vegetation of Southern Africa. Cambridge University Press, Cambridge, p. 615.

Richardson, D.M., Bond, W.J., Dean, W.R.J., Higgins, S.I., Midgley, G.F., Milton, S.J., Powrie, L., Rutherford, M.C., Samways, M.J., Schulze, R.E., 2000. Invasive alien organisms and global change: a South African perspective. In: Mooney, H.A., Hobbs, R.J. (Eds.), Invasive Species in a Changing World. Island Press, Washington D.C, pp. 303–349.

Richardson, D.M., Holmes, P.M., Esler, K.J., Galatowitsch, S.M., Stromberg, J.C., Kirkman, S.P., Pyšek, P., Hobbs, R.J., 2007. Riparian vegetation: degradation, alien plant invasions, and restoration prospects. Diversity and Distributions 13, 126–139.

Ruwanza, S., Gaertner, M., Esler, K.J., Richardson, D.M., 2013a. Both complete clearing and thinning of invasive trees lead to short-term recovery of native riparian vegetation in the Western Cape, South Africa. Applied Vegetation Science 16, 193–204.

Ruwanza, S., Gaertner, M., Richardson, D.M., Esler, K.J., 2013b. Soil water repellency in riparian systems invaded by *Eucalyptus camaldulensis*: a restoration perspective from the Western Cape Province, South Africa. Geoderma 200–201, 9–17.

Scott, D.F., Versfeld, D.B., Lesch, W., 1998. Erosion and sediment yields in relation to afforestation and fire in the mountains of the Western Cape Province, South Africa. South African Geographical Journal 80, 52–59.

StatSoft Inc, 2010. STATISTICA (data analysis software system). Version 10 http://www.statsoft.com.

Tererai, F., Gaertner, M., Jacobs, S.M., Richardson, D.M., 2013. *Eucalyptus* invasions in riparian forests: effects on native vegetation community diversity, stand structure and composition. Forest Ecology and Management 297, 84–93.

Van der Merwe, H., Van Rooyen, M.W., 2011. Vegetation trends following fire in the Roggeveld, Mountain Renosterveld, South Africa. South African Journal of Botany 77, 127–136.

Van Wilgen, B.W., Le Maitre, D.C., Cowling, R.M., 1998. Ecosystem services, efficiency, sustainability and equity: South Africa's working for water programme. Trends in Ecology & Evolution 13, 378.

Velempini, P., Riddoch, I., Batisani, N., 2003. Seed treatments for enhancing germination of wild okra (*Corchorus olitorius*). Experimental Agriculture 39, 441–447.

Vosse, S., Esler, K.J., Richardson, D.M., Holmes, P.M., 2008. Can riparian seed banks initiate restoration after alien plant invasion? Evidence from the Western Cape, South Africa. South African Journal of Botany 74, 432–444.

Wallis, M.G., Horne, D.J., 1992. Soil water repellency. Advances in Soil Sciences 20, 91–146.

Yelenik, S.G., Stock, W.D., Richardson, D.M., 2004. Ecosystem impacts of invasive *Acacia saligna* in South African fynbos. Restoration Ecology 12, 44–51.